A process for developing and evaluating indices of fish assemblage integrity

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Abstract: We describe a general process for developing an index of fish assemblage integrity, using the Willamette Valley of Oregon, U.S.A., as an example. Such an index is useful for assessing the effects of humans on entire fish assemblages, and the general process can be applied to any biological assemblage and any region. First, a reference condition was determined from historical information, and then candidate metrics of ecological importance were listed. The variability of the candidate metrics in time and space was estimated and their responsiveness to independent measures of riparian and stream habitat quality assessed. Metrics were scored continuously from 0 to 10, producing an index of biological integrity (IBI) that was weighted to range from 0 to 100 regardless of the number of metrics. The index, developed from a set of 35 sites, was then tested on an independent set of eight urban sites sampled by the Oregon Department of Fish and Wildlife. Thirteen of the 16 candidate metrics were appropriate and produced an IBI with among-site variance triple that of revisit variance. The method distinguished sites with acceptable fish assemblages from marginally and severely impaired sites.

Résumé : Nous décrivons un processus général pour l'élaboration d'un indice de l'intégrité des groupements de poissons, en utilisant la vallée de la Willamette, en Oregon, aux États-Unis, à titre d'exemple. Un indice de cette sorte est utile pour évaluer les effets des activités humaines sur des groupements entiers de poissons, et le processus général peut être appliqué à tout groupement biologique et dans n'importe quelle région. En premier lieu, nous avons déterminé un état de référence à partir d'informations historiques, puis nous avons dressé une liste de paramètres pertinents sur le plan écologique. Nous avons estimé la variabilité des paramètres dans le temps et dans l'espace et évalué leur réactivité à des mesures indépendantes de la qualité de l'habitat riverain et fluvial. Les paramètres étaient évalués sur une échelle continue allant de 0 à 10 pour produire un indice de l'intégrité biologique (IIB) qui a été pondéré de façon à s'étendre de 0 à 100, quel que soit le nombre de paramètres. L'indice, établi pour un ensemble de 35 sites, a ensuite été testé sur un ensemble indépendant de huit sites urbains échantillonnés par l'Oregon Department of Fish and Wildlife. Treize des 16 paramètres considérés étaient appropriés et ont produit un IIB ayant une variance inter-sites trois fois supérieure à la variance recalculée. Cette méthode a permis de repérer des sites abritant des groupements de poissons acceptables par rapport aux sites peu ou très dégradés.

[Traduit par la Rédaction]

Introduction

Biological integrity, an objective of the 1972 U.S. Clean Water Act and Canada's National Park Act, has been defined as the ability to support and maintain a balanced, integrated, adaptive community of organisms having a composition, diversity, and functional organization comparable with that of the natural habitats of the region (Frey 1977). We consider a community as having integrity if its composition and function are comparable with those of natural habitats, which we define as minimal human disturbance, or pre-Columbian conditions. Such conditions are only benchmarks for comparison and may not be achievable given current land uses.

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Biological integrity is best monitored at the community or assemblage level because indicator species or populations are insufficiently robust indicators (Landres 1992). We agree with Fauth et al. (1996) and use the term "assemblage" to represent a phylogenetic subset of a community, while a community is the entire biological component of an ecosystem. A widely used indicator of fish assemblage integrity is the index of biological integrity (IBI, Karr et al. 1986). Initially developed for midwestern U.S. streams, it has been modified for streams of Appalachia, Ontario, North Carolina, Colorado, Tennessee, Idaho, Missouri, and Mexico, large rivers in Oregon, France, Ohio, Australia, Africa, Belgium, and India, Tennessee river reservoirs, and Great Lakes bays (Simon and Lyons 1995; Hughes and Oberdorff 1998). However, few applications included evaluations of metric and IBI variability and responsiveness to disturbance; Minns et al. (1994) is a notable exception.

IBIs reflect important components of ecology: taxonomic richness, habitat and trophic guild composition, and individual health and abundance. Metrics in each category are scored and then summed into an IBI value. Differences in expected fish species richness and composition associated with different regions or basins, water body sizes, and location in a drainage are factored into metric selection and scoring (Fausch et al. 1984; Miller et al. 1988). IBIs focus on fish assemblage structure rather than ecosystem processes because the latter appear less responsive to stressors (Schindler 1990). As recommended by Karr et al. (1986), however, both structural and functional metrics are included. IBIs translate an aquatic ecologist's assessment of fish assemblage integrity for persons unfamiliar with fishes. They alone cannot convey causal relationships or fundamental ecological processes.

Our objectives were to demonstrate a general method for developing and evaluating biological indicators of biological integrity that can be useful for monitoring across large regions for long time periods. As an example, we describe how we determined reference conditions, developed an IBI for Willamette Valley, Oregon, U.S.A., fish assemblages, assessed the variance of IBI metrics, compared IBI scores with physical and chemical habitat, and tested the IBI on an independent data set.

Methods

Reference condition

Commonly, natural condition is estimated from minimally disturbed sites, but in heavily altered regions, such sites are absent. In those cases, historical data, paleoecological data, quantitative models, and expert judgement are used to define the natural condition (Hughes 1995). Willamette Valley streams are widely disturbed by agriculture and urbanization. Therefore, we based the reference condition on estimates of pre-Columbian hydrological and stream habitat characteristics, fish–habitat relationships, and the regional fish species pool as proposed in Hughes (1995). These were developed from an evaluation of historical museum data (Hughes et al. 1987), nine summers of stream sampling throughout the Valley, conversations about past stream and fish assemblage conditions with emeritus professors of ichthyology (R. Dimick and C. Bond, Oregon State University, Corvallis, OR), and reviews of historical conditions (Johannessen et al. 1971; Benner and Sedell 1997). We developed candidate IBI metrics based on our interpretation of historical conditions and the habitat and trophic requirements of the Valley's fish species (Table 1). Fausch et al. (1984) used a similar approach for species richness and guild types of metrics across a greater range of stream sizes.

Candidate metrics

Following the lead of Karr et al. (1986), we identified candidate metrics that represent major aspects of fish assemblage integrity: taxonomic richness, habitat guilds, trophic guilds, and individual health and abundance. To increase resolution among sites and reduce variance, we also modified their scoring criteria (discussed below).

Taxonomic richness

Karr et al. (1986) focused on species richness. We substituted native species richness as they recommended for streams of the western United States. We did not want to confound the species richness metric, which frequently decreases with degradation, with alien species, which represent biological degradation and homogenization and often increase with disturbance. We also added a metric, native family richness, to assess an additional level of biological diversity. Oberdorff and Hughes (1992) reported that entire families were extirpated from, or threatened in, basins with long and intensive human occupation.

Habitat guilds

Karr et al. (1986) used the number of darter, sunfish, sucker, and intolerant species to evaluate fish assemblage components likely to decrease in response to habitat degradation. Because darters and sunfish are alien to Oregon, we followed the suggestion of Karr et al. (1986) and Miller et al. (1988) and used native species richness of benthic and water column fishes instead. Habitats for each species (Table 1) were determined from Scott and Crossman (1973), Moyle (1976), Becker (1983), and Bond et al. (1998). Suckers only rarely occurred in Willamette Valley streams; a sucker metric would therefore be insensitive, so we did not include one. In place of intolerant species richness, we used sensitive species richness and the species richness of nonguarding lithophil nesters. These species are early-warning indicators of anthropogenic disturbance and they rarely occur in highly turbid, warm, chemically polluted, or heavily silted streams. They occur most often in clear, cool, unpolluted streams with complex cover and coarse substrates (although they tolerate extreme flows, cold temperatures, and oligotrophic conditions). Oberdorff and Hughes (1992) also used a lithophil metric. Because such species were rarely abundant in even the least disturbed Willamette Valley streams, we based these metrics on species richness rather than proportions of individuals of these species in the assemblage. We also evaluated native hider species richness. Hiders (Table 1) are fishes commonly found in substrate interstices or among macrophytes or organic debris. Because they were difficult to collect and rarely abundant, we again based the metric on the number of hider species rather than the proportion of individuals collected that are hiders.

To assess assemblage components that increase with habitat disturbance, we substituted percent alien (nonnative) and percent tolerant individuals in the sample for the percent green sunfish metric originally used by Karr et al. (1986). The percent aliens reflects biological pollution, which is usually less reversible than chemical and physical disturbance; it was also the second most commonly cited cause of fish species extinction after physical habitat change (Miller et al. 1989). Karr et al. (1986) and Miller et al. (1988) also suggested substituting percent tolerants for percent green sunfish. Tolerants (Table 1) were determined from Scott and Crossman (1973), Moyle (1976), and Becker (1983); such species are generally successful in streams with high silt loads and warm, turbid, and poorly oxygenated water. All tolerants in our streams were aliens, although not all the aliens were considered tolerant. Several commonly occurring native species were not included as tolerants because they are are less tolerant than the aliens and can be very abundant in minimally disturbed systems. We used percent tolerant individuals in the sample versus tolerant species richness, which helped us assess the tendency for only one of these species to dominate in highly disturbed situations or where few individuals persisted.

Trophic guilds

Karr et al. (1986) used percent individuals as omnivores, insectivores, and piscivores to evaluate the status of the trophic structure and food base. We adopted the omnivore metric, but rejected the other two. Omnivores typically eat substantial amounts of both plant and animal material, allowing such species to adapt easily to disrupted food sources. There are very few true piscivores in Willamette River basin streams and most individuals were invertivores. This meant that regardless of their relative integrity, streams lacked piscivores. Invertivores, on the other hand, were very abundant in all streams and did not demonstrate marked differences in proportions as habitat quality differed, which resulted in unresponsive metrics. We substituted percent filter feeders and percent native top carnivores for the invertivore and piscivore metrics. Filter feeders (juvenile lampreys) are sensitive to migratory barriers, suspended sediment concentration, and the quality of fine particulate organic matter and microorganisms in the drift and bedload. The top carnivore metric evaluates the ability of the stream to produce enough fish and large invertebrates to support relatively large native predators. It was suggested as a piscivore substitute by Karr et al. (1986). We used Scott and Crossman (1973), Moyle (1976), Wydoski and Whitney (1979), Becker (1983), and Li et al. (1987) to designate trophic guilds for each species (Table 1).

Individual health and abundance

The total number of individuals and percent individuals that are hybrids or diseased were used by Karr et al. (1986) to assess stream

Table 1. Fishes of wadeable streams in the Willamette Valley ecoregion.

Family/species	Origin[a]	Habitat[b]	Tolerance[c]	Foraging[d]	Reproduction[e]
Petromyzontidae					
Lampetra richardsoni	N	BH	S	F/S	NLN
Lampetra tridentata	N	BH	S	F/S	NLN
Cyprinidae					
Acrocheilus alutaceus	N	B	I	S/S	L
Carassius auratus	A	B	T	O	V
Ctenopharyngodon idella	A	B	T	H	V
Cyprinus carpio	A	B	T	O	V
Mylocheilus caurinus	N	W	I	I	L
Oregonichthys crameri	N	WH	S	I	V
Ptychocheilus oregonensis	N	W	I	T	L
Pimephales promelas	A	B	T	O	P/CN
Rhinichthys cataractae	N	BH	I	I	L
Rhinichthys falcatus	N	BH	I	I	L
Rhinichthys osculus	N	BH	I	I	LN
Richardsonius balteatus	N	W	I	I	LV
Cobitidae					
Misgurnus anguillicaudatus	A	B	T	O	V
Catostomidae					
Catostomus macrocheilus	N	B	I	O	L
Catostomus platyrhynchus	N	B	S	S/S	L
Ictaluridae					
Ameiurus melas	A	BH	T	O	P/CN
Ameiurus natalis	A	BH	T	O	P/CN
Ameiurus nebulosus	A	BH	T	O	P/CN
Salmonidae					
Oncorhynchus clarki	N	WH	S	T	NLN
Oncorhynchus kisutch	N	W	S	T	NLN
Oncorhynchus mykiss	N	WH	S	T	NLN
Oncorhynchus tshawytscha	N	W	S	T	NLN
Prosopium williamsoni	N	B	S	I	NLN
Percopsidae					
Percopsis transmontana	N	BH	I	I	L
Fundulidae					
Fundulus diaphanus	A	W	T	O	V
Poeciliidae					
Gambusia affinis	A	WH	T	O	LB
Gasterosteidae					
Gasterosteus aculeatus	N	WH	I	I	VN
Cottidae					
Cottus asper	N	B	I	I	CN
Cottus beldingi	N	BH	S	I	CN
Cottus gulosus	N	BH	I	I	CN
Cottus perplexus	N	BH	I	I	CN
Cottus rhotheus	N	BH	S	T	CN
Centrarchidae					
Lepomis auritus	A	W	T	I	PN
Lepomis cyanellus	A	W	T	T	PN
Lepomis gibbosus	A	W	I	I	PN
Lepomis gulosus	A	W	T	T	PN
Lepomis macrochirus	A	W	T	I	PN
Lepomis microlophus	A	W	T	I	PN
Micropterus dolomieu	A	W	I	T	LN
Micropterus salmoides	A	W	T	T	PN
Pomoxis annularis	A	W	T	T	VN

Table 1 (*concluded*).

Family/species	Origin[a]	Habitat[b]	Tolerance[c]	Foraging[d]	Reproduction[e]
Pomoxis nigromaculatus	A	W	T	T	PN
Percidae					
Perca flavescens	A	W	I	T	V

Note: Data from Scott and Crossman (1973), Moyle (1976), Wydoski and Whitney (1979), Becker (1983), Li et al. (1987), and Bond et al. (1988).

[a]N, native; A, alien.

[b]B, benthic; W, water column; H, hider.

[c]S, sensitive; I, intermediate; T, tolerant.

[d]F/S, filterer/specialist; S/S, scraper/specialist; O, omnivore; I, invertivore; T, top carnivore.

[e]NLN, nonguarding lithophil (gravel–cobble) nester; LN, lithophil nester; L, lithophil; V, vegetation; P, psammophil (sand – fine gravel); CN, cavity nester; LB, livebearer; VN, vegetation nester; PN, psammophil nester.

productivity and individual fitness. The hybrid metric was excluded because hybrids were rare, or we lacked the ability to recognize them. We included a metric for percent individuals that were diseased or had external anomalies. Typically, such an anomaly metric is most useful with larger fish and in waters that are severely degraded by chemical contaminants. We also examined a "total number of individuals" metric, although both oligotrophic and highly disturbed physical or chemical habitats typically support exceedingly low fish numbers. To evaluate the effects of degradation on long-lived individuals, we examined the percent individuals that were native adults and the percent long-lived native species with individuals reaching large sizes ("lunkers," Table 2). These two metrics also assess flow permanence and habitat suitability as well as the site's ability to support adults. Johnson (1994) reported that minimally disturbed waters were frequently dominated by large, old individuals.

Metric and index scoring

Karr et al. (1986) scored metrics as 5, 3, or 1 if their data approximated, deviated slightly from, or were markedly different from reference conditions. Thus the 12 metrics they used produced an index ranging from 12 to 60 if fish were present. We agree with Minns et al. (1994) that an index based on continuous scoring of 0.0–10.0 for metrics and 0.0–100.0 for the IBI is preferable. The decimal system is a familiar one, as are ranges from 0 to 10 or 100. Also, continuous scoring, including decimal fractions, reduces variances when metric values differing by a value ≤ 1 are scored as different categories. These changes, by offering more accurate depiction of the data, should make IBIs less variable and more easily understood.

We set the upper and lower thresholds for each metric based on our interpretation of pre-Columbian stream conditions, species ranges, fish habitat requirements, and reference sites (Table 2). Upper thresholds (scores of 10) vary with stream size and among metrics. Lower thresholds (scores of 0) were 0 for all metrics except percent alien, percent tolerant, percent omnivores, and percent anomalies; these were set at 10, 10, 10, and 2, respectively. These latter four metrics received higher metric scores as data values decreased to 0. Scores between the upper and lower thresholds were calculated by linear interpolation (dividing the metric value by its range and then multiplying by 10) as described by Minns et al. (1994). For example, the maximum expected value for the hider metric was four species, so three hider species received a score of 7.5. IBI scores are the sums of the metric scores times 10 and divided by the number of metrics, producing a maximum IBI score of 100 regardless of the number of metrics selected in future applications. Minns et al. (1994) also used this approach.

Table 2. Scoring criteria for IBI metrics used in order 1–3 streams.

Metric	Raw values: Stream order 1	Raw values: Stream orders 2 and 3
Taxonomic richness		
No. of native families	0–4	0–7
No. of native species	0–5	0–11
Habitat guilds		
No. of native benthic species	0–3	0–7
No. of native water column species	0–2	0–4
No. of hider species	0–4	0–4
No. of sensitive species	0–2	0–5
No. of native nonguarding lithophil nester species	0–3	0–3
% tolerant individuals	10–0	10–0
% alien individuals[a]	10–0	10–0
Trophic guilds		
% filter-feeding individuals	0–10	0–10
% native top carnivore individuals	0–10	0–10
% omnivores	10–0	10–0
Individual health and abundance		
% of target species that include lunkers[b]	0–100	0–100
% individuals with anomalies	2–0	2–0
% native species with adult individuals[a]	0–100	0–100
Total no. of individuals[a]	0–50	0–50

Note: Raw data values at the low (and lower) end of the ranges are scored as 0; those at the high (and higher) end are scored as 10; scores for intermediate values are calculated by dividing the raw value by the score range (see text for examples). Strahler stream orders determined from 1 : 100 000 scale maps.

[a]Metric not included in final IBI.

[b]Lunkers are relatively old large individuals of the following species and sizes: *C. asper* (10 cm), *C. rhotheus* (10 cm), *O. clarki* (25 cm), *O. mykiss* (30 cm), *A. alutaceus* (30 cm), *P. oregonensis* (30 cm), *C. macrocheilus* (30 cm). See Table 1 for full species names.

Data sources

Our data originated from a set of 21 subjectively chosen stream sites sampled three times in the summer of 1982 and 18 randomly selected stream sites sampled in 1993. Eight of the latter 18 streams were also sampled twice in the summers of 1992–1995 to assess variance components (Herlihy et al. 1997). All streams were in the Willamette River basin and the Willamette Valley ecoregion (Omernik 1987). Data from randomly selected reaches ensured that data were representative of most Valley streams, while data from the subjectively chosen sites ensured that extremes in stream quality were represented. Stream reaches were sampled for fish with backpack electrofishers during the summer base flow period, but effort varied from three passes on 100 m in 1982 to one pass on reaches 40 times as long as their wetted channel widths in 1992–1995 (McCormick and Hughes 1997). A separate set of eight subjectively chosen sites electrofished by three passes of 100 m in 1993 and 1994 (Friesen and Ward 1996) was used as an independent test of the IBI.

Water quality samples and physical habitat data were taken during each visit to the randomly selected reaches (Herlihy et al. 1997; Kaufmann and Robison 1997). Physical habitat was sampled quantitatively by two persons for 3–4 hours. It included a count of the large woody debris and 100 measurements of depth along the thalweg. In addition, at 11 cross sections, measurements of width, bank height and angle, gradient, sinuosity, and riparian canopy cover were taken, and substrate size, embeddedness, riparian vegetation condition, fish cover, and human disturbances were visually estimated. A 4-L cubitainer of stream water was collected in a flowing portion near the middle of the stream for major anion, cation, conductivity, and nutrient analyses. Detailed information on the analytical and preservation procedures for each analyte can be found in USEPA (1987). In brief, major anions (sulfate, nitrate, chloride) were determined by ion chromatography, base cations by atomic absorption, and total nitrogen and phosphorus by persulfate oxidation and colorimetry. Dissolved oxygen (DO) concentrations for the Tualatin River basin (an urbanized subbasin of the Willamette) sites were provided by the Unified Sewerage Agency. Those data were collected weekly in the summers of 1990–1994 between 08:00 and 09:50 through use of daily calibrated DO meters.

Data analyses

We determined if the metrics were sensitive at high, low, and intermediate levels of biological integrity, a desirable trait for an accurate and useful IBI. Initially, we sorted IBI and metric scores to help discern ranges and patterns; this led to modifications of scoring criteria by stream order for five metrics. Next, we plotted metric scores and raw metric values against IBI.

To eliminate metrics that were redundant and those that consistently masked the signal of other metrics, we calculated correlation coefficients for index and metric scores. We anticipated high positive correlations between the IBI and individual metrics and among the metrics, yet no identical correlations between metrics and the IBI. The sorted IBI and metric scores and the correlations aided us in finding metrics that behaved in a different manner than the others (consistent negative correlations). Cronbach's alpha (Cronbach 1951) and a metric-remainder correlation coefficient were also used. The metric-remainder coefficient was determined by calculating the IBI without the metric and then calculating the correlation between this modified IBI and the metric value. A metric that was concordant with the others had a large, i.e., close to 1, positive coefficient. Cronbach's alpha was determined by dividing the sum of all pairwise metric covariances by the total IBI variance, which included both metric variances and covariances. For a metric consistent with the others, its alpha will decrease relative to the alpha value for the IBI. Conversely, for a metric inconsistent with the others, its alpha will increase relative to the IBI alpha value. Cronbach (1951) recommended that alpha should be around 0.7 for a useful metric or index. This value signals a suitable level of redundancy, yet reduces opportunities for some metrics to consistently counter the scores of other metrics.

IBI and metric variability were evaluated in three ways. First, we plotted IBI and metric scores obtained from multiple samplings of the same sites against each other in 1:1 plots. The precision of the IBI in such plots was assessed by the proximity of the points to a 45° line. These plots also offered a simple visual means to determine if variability was greater for high or low IBI scores. Second, analysis of variance was used to compare the relative contributions of major sources of variance (Larsen et al. 1995). These included among-site (the variance resulting from differences among all sites sampled), year-to-year (the degree that all sites trend in the same direction in a particular year), individual site–year interaction (the amount a site varied from year to year around its mean or trend), and residual or error variance (variance contributed by measurement methods, crew-to-crew differences, and temporal variance during the sampling season). We also determined the ratio of among-site variance to the sum of all other sources of variance for each metric and the IBI. Third, we determined power curves for three significance levels at various power values and differences in IBI scores. Power, the ability of the IBI to detect differences among samples, was evaluated through use of the variance components determined above. In particular, we wanted to know for a power of 0.8 and an α of 0.05 (*i*) the detectable difference in mean IBI scores from one year to the next and (*ii*) the number of years of data needed to detect a 2% per year trend in mean IBI score.

We tested IBI responsiveness with a set of quantitative physical and chemical habitat data from the 18 randomly selected sites. We applied principal components analysis (PCA) to 16 chemical and physical habitat variables to express an integrated measure of habitat quality against which we evaluated the candidate fish metrics and IBI.

We also applied PCA to the fish data to evaluate a multivariate measure of fish assemblage integrity. We used PCA of raw metric values and scored metrics and plotted the first two components to determine if PCA could discriminate different degrees of fish assemblage integrity. In other words, were sites with markedly different habitat quality located apart on the graph and separated by those of intermediate quality?

The proportion of the sampled sites that had acceptable, marginally impaired, or severely impaired fish assemblage integrity was estimated by tabulating IBI scores, or mean IBI scores from reaches with multiple samples. The scores were then compared with the maximum potential IBI score (100). Scores were considered acceptable, marginally impaired, or severely impaired if they were ≥ 75, 51–74, or ≤ 50, respectively. The upper scoring break was comparable with Karr et al.'s (1986) break between good and fair categories, while the lower break was close to Karr et al.'s 57% cutoff for poor sites. Scores were also evaluated by comparison with the maximum observed mean IBI score (90) at a reach. In this case, acceptable, marginally impaired, or severely impaired scores were ≥ 68, 46–67, or ≤ 45, respectively.

Results

Metric performance

Most scored metrics (versus raw metric values) were positively and highly correlated ($r > 0.5$) with IBI, and most correlations among metrics were positive. Total number of individuals, percent large fish, percent anomalies, and percent native adults had low correlations with IBI and all other metrics. All seven richness metrics were highly correlated with each other, even benthic and water column species richness. The percent alien and percent tolerant metrics were redundant in this data set ($r = 1$). Similarly, most metric-remainder correlation coefficients were high for the total data set, except for total number of individuals, percent anomalies, and percent native adults (all with $r \leq 0.3$, second to last column of Table 3). Cronbach's alpha scores for the total data set were all positive and = 0.7 if rounded (last column of Table 3). These results indicate a relatively high degree of concordance among metrics, especially for the longer reaches sampled in 1992–1995.

A multivariate (PCA) representation of physical and chemical habitat quality was calculated independently through use of quantitative data from the 1992–1995 sites (Table 4). Low values of habitat PCA-1 (HPCA-1) were associated with high levels of human disturbance at the site, as well as high chloride and sulphate concentrations. High HPCA-1 scores occurred

Table 3. Values of the metric-remainder correlation coefficient for each scored metric and Cronbach's alpha for the IBI and for the restricted IBI associated with each of the scored metrics.

	1982, stream orders 2 and 3		1992–1995 Stream order 1		1992–1995 Stream orders 2 and 3		1982–1995, all three data sets	
	Coeff.	Alpha	Coeff.	Alpha	Coeff.	Alpha	Coeff.	Alpha
IBI		0.33		0.73		0.66		0.65
Native family richness	0.40	0.40	0.98	0.73	0.87	0.67	0.81	0.67
Native species richness	0.53	0.39	0.93	0.73	0.90	0.67	0.86	0.66
Hider species richness	0.72	0.33	0.91	0.74	0.49	0.69	0.76	0.66
Benthic species richness	0.44	0.40	0.81	0.74	0.84	0.67	0.79	0.67
Water column species richness	0.41	0.39	0.80	0.76	0.34	0.70	0.65	0.67
Sensitive species richness	0.82	0.34	0.83	0.74	0.70	0.68	0.84	0.66
Nonguarding lithophil nester richness	0.75	0.32	0.97	0.73	0.73	0.67	0.85	0.65
Total no. of individuals	0.06	0.44	0.68	0.75	0.47	0.69	0.24	0.70
% filter-feeding individuals	0.54	0.36	0.68	0.75	0.74	0.67	0.68	0.67
% alien individuals	0.62	0.33	0.35	0.77	0.68	0.68	0.54	0.68
% native top carnivore individuals	0.47	0.37	0.78	0.74	0.53	0.69	0.60	0.68
% tolerant individuals	0.62	0.33	0.31	0.77	0.69	0.68	0.55	0.68
% target species with lunkers	0.21	0.41	0.25	0.77	0.48	0.69	0.35	0.69
% individuals with anomalies	–0.01	0.49	—[a]	0.77	—[a]	0.71	0.20	0.71
% omnivorous individuals	0.13	0.47	0.31	0.77	0.54	0.69	0.37	0.70
% native adult individuals	0.01	0.43	0.45	0.76	0.38	0.70	0.29	0.70

[a]Values of the metric were constant for this data set.

where riparian vegetation was abundant and multilayered, instream fish cover was common and diverse, and residual pool volume was high.

Raw metric data from the 1993 samples demonstrated expected, but frequently weak and variable, patterns when plotted against HPCA-1 scores. The limited number of points often did not indicate strong associations between metric data and HPCA scores (Fig. 1). Three metrics (percent alien, percent tolerant, percent omnivores) showed high values only at sites with low habitat quality. A set of four metrics (native nonguarding lithophil nesters, percent filter feeders, percent native top carnivores, percent species with large individuals) showed the opposite pattern: high values only occurred when habitat quality was high. Data for percent native adults appeared unrelated to habitat quality. The remaining metrics increased constantly with increased habitat quality. The candidate metrics, therefore, included some that appeared sensitive at either high or low extremes of habitat quality as well as those responsive across the range of habitat quality expected in most Willamette Valley streams. When all the metrics were scored, combined into an IBI, and plotted against habitat quality, there was a significant ($r = 0.65$, $p < 0.01$) positive relationship.

Because of small sample size and considerable variability, Fig. 1 provided weak evidence for metric responses to habitat quality. To the degree that IBI is also a measure of ecological integrity and habitat quality (as suggested by the highly significant relationship between the two), we can examine patterns of raw metric scores plotted against IBI scores for the combined 1982 and 1992–1995 data (Fig. 2). All but the total number of individuals and the percent native adult metrics showed the expected patterns.

As a result of these analyses, three metrics (total number of individuals, percent native adults, percent alien) were removed from the final IBI. Total number of individuals and percent native adults were unresponsive to general habitat disturbance and the IBI score, while percent alien and percent tolerant were identical. A fourth metric, percent anomalies, was retained because of insufficient data for rejection.

Table 4. Eigenvectors and accountable variances of the first two principal components based on physical and chemical habitat variables.

PCA-1	PCA-2	Variable
0.84	0.32	Mean woody cover (ground, midlayer, canopy)
0.82	0.28	Mean canopy cover in riparian zone
0.79	0.09	Mean large woody debris in flooded channel
0.76	0.22	Mean small woody debris in flooded channel
0.71	–0.05	Mean width × depth
–0.66	0.01	Anthropogenic disturbances within 10 m of stream
0.58	–0.08	Residual pool area
0.53	–0.17	Standard deviation of width
–0.63	0.43	Sulphate
–0.58	0.46	Conductivity
–0.58	0.34	Chloride
–0.31	0.66	Total nitrogen
–0.21	0.36	Mean vegetation cover overhanging stream
0.25	0.86	% fines and sand
–0.28	–0.83	Mean substrate size
37	18	% variance accounted for

PCA of IBI metrics

PCA of species abundance data did not produce interpretable patterns, but a PCA using IBI metrics (BPCA) did. The first and second principal component factors of metric scores accounted for 44 and 13% of metric score variability, respectively (Table 5). All 13 metrics were positively associated with BPCA-1, which further indicated internal consistency among the IBI metrics. The richness metrics loaded more highly

Fig. 1. Raw metric scores versus physical and chemical habitat quality for 15 metrics. Habitat quality was estimated from PCA-1 (see Table 4) and increases from left to right. Metrics showed constantly increasing, rapidly decreasing, slowly increasing, or highly variable responses to increasing habitat quality.

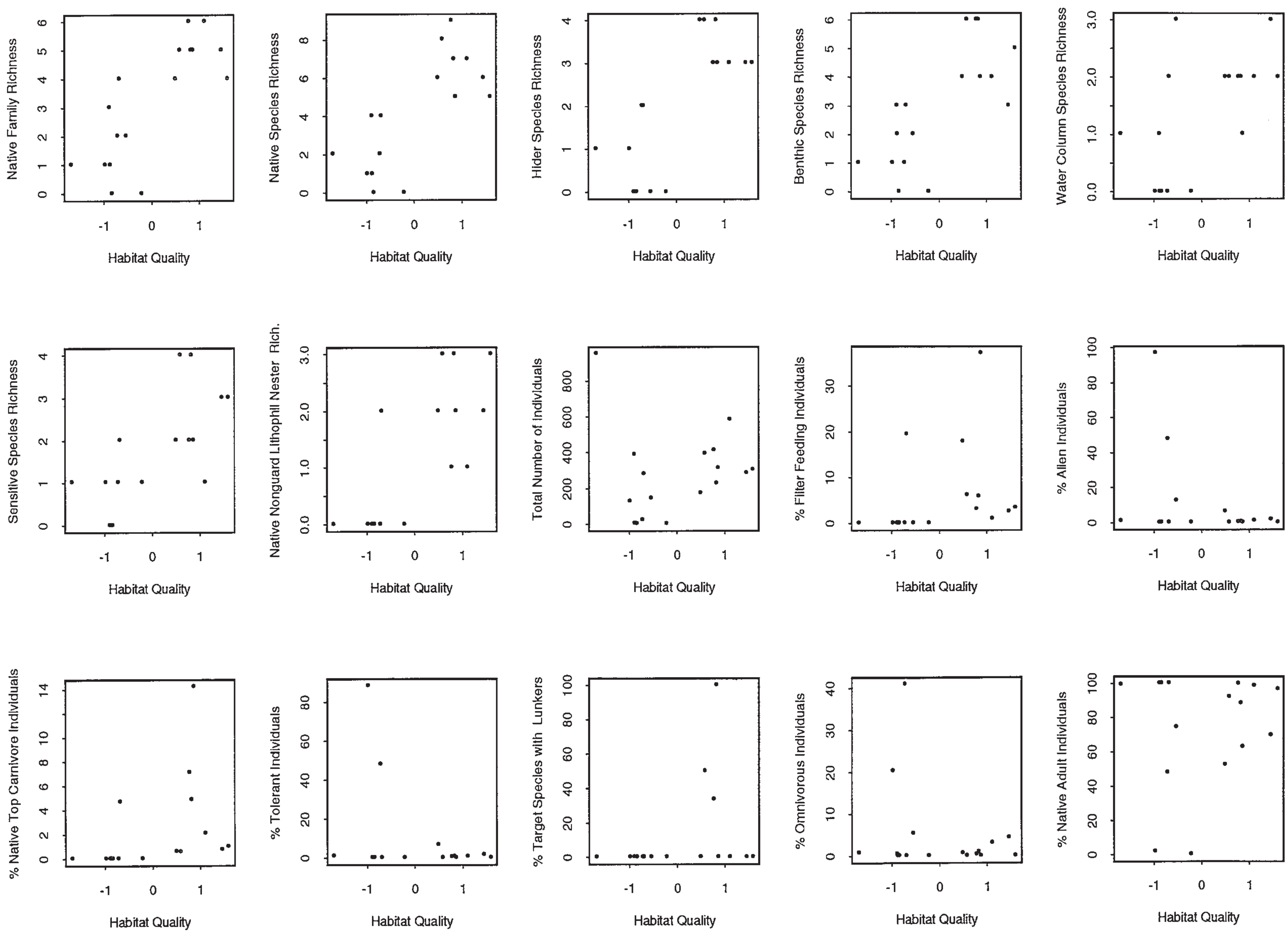


Fig. 2. Raw metric scores versus IBI scores. Metrics showed constantly increasing, rapidly decreasing, slowly increasing, or highly variable responses to increasing IBI scores.

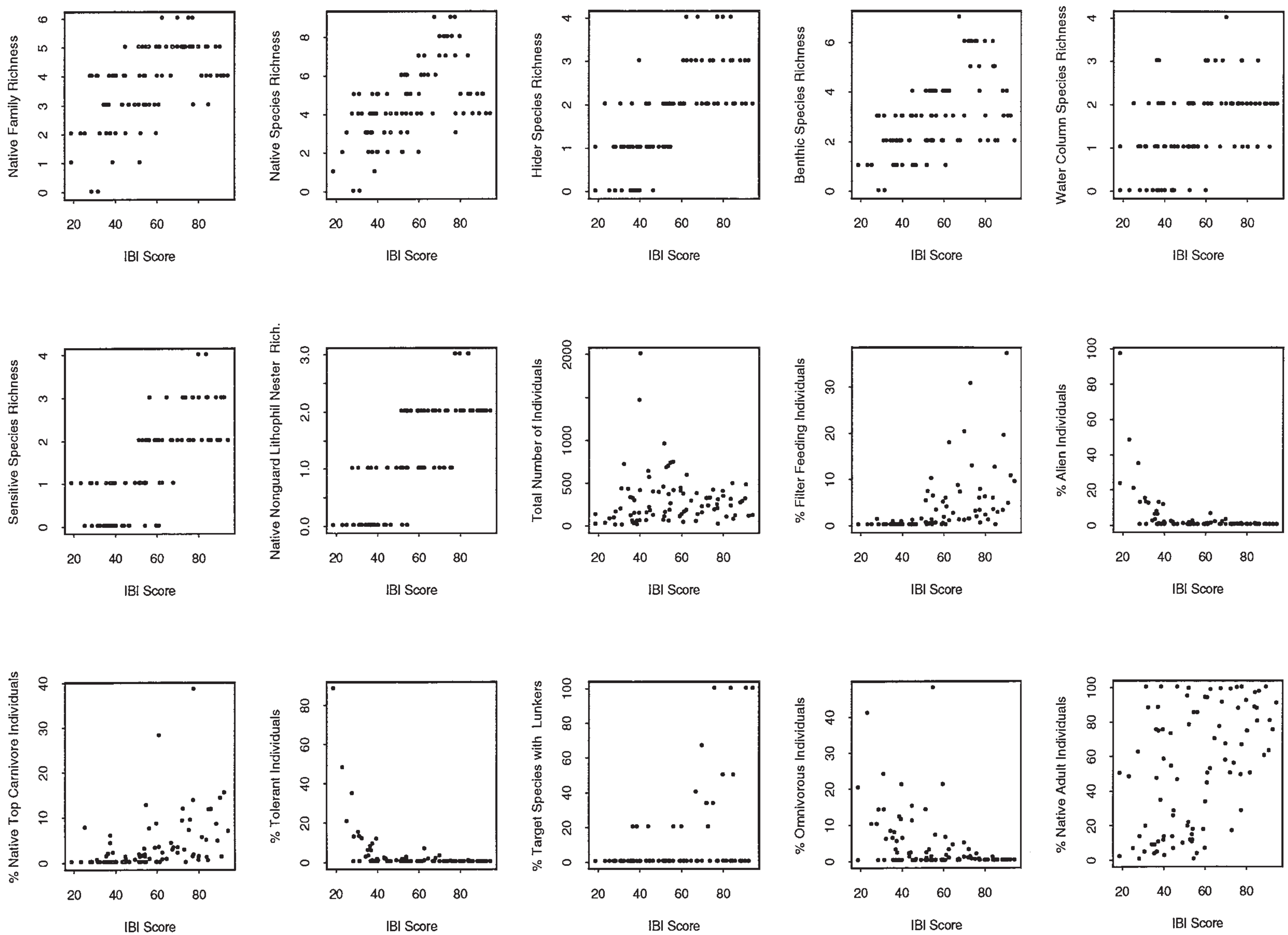


Table 5. Eigenvectors and accountable variances of the first two principal components based on IBI metric scores.

PCA-1	PCA-2	IBI candidate metric
0.34	–0.18	Native fish family richness
0.35	–0.15	Native fish species richness
0.32	–0.16	Hider species richness
0.32	–0.06	Native benthic species richness
0.28	–0.18	Water column species richness
0.33	–0.04	Sensitive species richness
0.34	–0.05	Nonguarding lithophil nester richness
0.13	–0.27	Total no. of individuals
0.23	0.04	% filter-feeder individuals
0.19	0.52	% alien individuals
0.21	–0.06	% top carnivore individuals
0.19	0.52	% tolerant individuals
0.16	0.03	% target species with lunkers
0.09	–0.05	% individuals with anomalies
0.15	0.49	% omnivorous individuals
0.15	0.17	% native adult individuals
44	13	% variance accounted for

(0.28–0.35) than the percent metrics (0.09–0.23). These results suggest that BPCA-1 may be considered an IBI. This was also indicated by the close association of IBI and BPCA-1 scores ($r = 0.99$). BPCA-2 correlated most with the IBI metrics indicating disturbance (percent tolerant, percent alien, percent omnivores). When BPCA-1 and BPCA-2 were plotted together, sites with low scores on both axes (lower left of Fig. 3) had few omnivorous or tolerant (alien) species, yet had low IBI scores. There was also a negative relationship between BPCA-1 and BPCA-2 for small (first-order) streams, indicating that omnivorous or tolerant (alien) species declined as IBI increased in these streams. PCA of raw metric values (not shown) gave similar results. These findings indicated that IBI metrics ranked fish assemblage integrity whether aggregated through use of a multimetric index or a multivariate analysis. Successful extraction of this pattern, however, was based on analysis of a set of ecologically meaningful metrics, not the typical ordination of species and abundances.

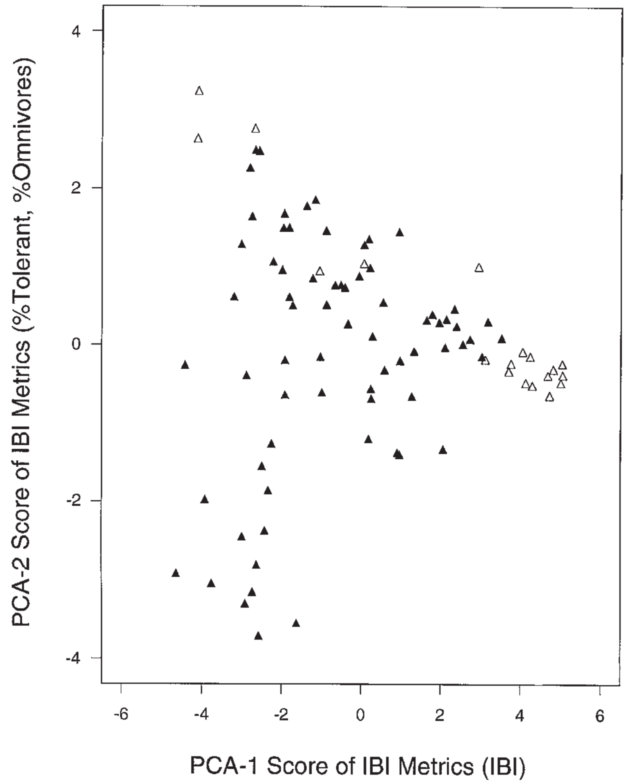
IBI variance and power

Variances were estimated for the 1992–1995 data, which included both interannual and intraseasonal repeat sampling. Among-site variance comprised the preponderance of total variance in IBI, with considerable stream/year interaction variance, and much smaller among-year variance and residual variance (Table 6). This is also true for most individual metrics, except for percent filter feeders, percent top carnivores, percent lunkers, and percent omnivores. Fore et al. (1994) found similar patterns in sources of variance for IBI.

A simple graphic of variance was obtained by plotting the IBI results of repeat sampling against the mean IBI score of multiple samples collected at the site in 1982 (Fig. 4). Similar plots were examined for seven sites sampled multiple years (not shown). In both cases, and as indicated by Table 6, temporal differences in IBI scores were typically less than 10% (10 units) of the potential IBI maximum and substantially lower than differences among sites. When the standard deviations of IBI scores were plotted against mean IBI scores for sites with repeat samples, low IBI scores reflected lower standard deviations than intermediate or high scores, although the pattern was weak (Fig. 5).

Statistical power analyses indicated that our IBI, based on a sample size of 18 streams each sampled once, can detect an 8-unit (8%) year-to-year difference in mean IBI at a power of 0.8 and $\alpha = 0.05$ (Fig. 6A). A 2-unit (2%) per year trend is detectable in 5 years (Fig. 6B).

Fig. 3. First and second principal component scores from IBI metrics (open triangles, first-order streams; solid triangles, second- and third-order streams). Stream sites in the lower left had few omnivorous and tolerant (alien) individuals but low IBI scores.



Independent validity test and impairment assessment

Another test of the usefulness of an IBI is its performance with an independently collected and assessed data set. Our IBI, calculated from 1994 data provided by Friesen and Ward (1996), was highly correlated ($r = 0.85$) with DO concentration.

The proportion of all sites that were in acceptable, marginally impaired, or severely impaired condition depended on the evaluation criteria. Depending on whether the maximum potential IBI score (100) or the maximum actual score (90) was used, 84–91% of sites were considered marginally or severely impaired, while 9–16% were considered in acceptable condition. These percentages reflected the biological criteria and the site selection process (both subjectively chosen and randomly selected), as well as the extent of landscape disturbance in the Willamette Valley.

Table 6. IBI and individual metric variances for 1992–1995.

	Among-site	Year	Stream × year	Residual	Ratio[a]
Family richness	6.15	–0.03	0.72	1.07	3.44
Species richness	5.27	0.12	1.15	0.84	2.50
Hider richness	9.52	0.09	0.39	1.94	3.93
Benthic richness	8.13	0.64	0.59	0.62	4.39
Water column richness	5.33	0.63	2.20	2.50	1.00
Sensitive species richness	9.62	0.03	5.21	0.70	1.62
Nonguarding lithophil nester richness	12.36	–0.53	1.91	2.78	2.64
% filter-feeding individuals	3.28	–0.81	3.59	5.73	0.35
% top carnivore individuals	3.26	3.66	7.55	0.66	0.27
% tolerant individuals	8.29	0.32	2.32	0.99	2.28
% target species with lunkers	2.84	0.00	–4.48	11.53	0.25
% individuals with anomalies	—	—	—	—	—
% omnivorous individuals	4.55	0.26	3.46	1.27	0.91
IBI	374.87	–16.89	98.65	26.13	3.00

[a]Among-site variance divided by sum of all remaining positive terms. By definition, variance components are positive, but negative variances can occur when variance is close to zero. We view the negative estimate as evidence that the true value of that component approximates zero.

Fig. 4. IBI precision as indicated by 1 to 3 monthly sampling visits at the same site in a summer. Numbers represent months sampled (1 = June, 2 = July, 3 = August), and their vertical positions indicate the IBI score on that visit versus the mean of all visits to that site. Mean IBI scores among sites ranged from 25 to 71, while the ranges of IBI scores at a site were usually <10. Also, low mean IBI scores (<40) varied less from the 1:1 diagonal line than did intermediate and higher values.

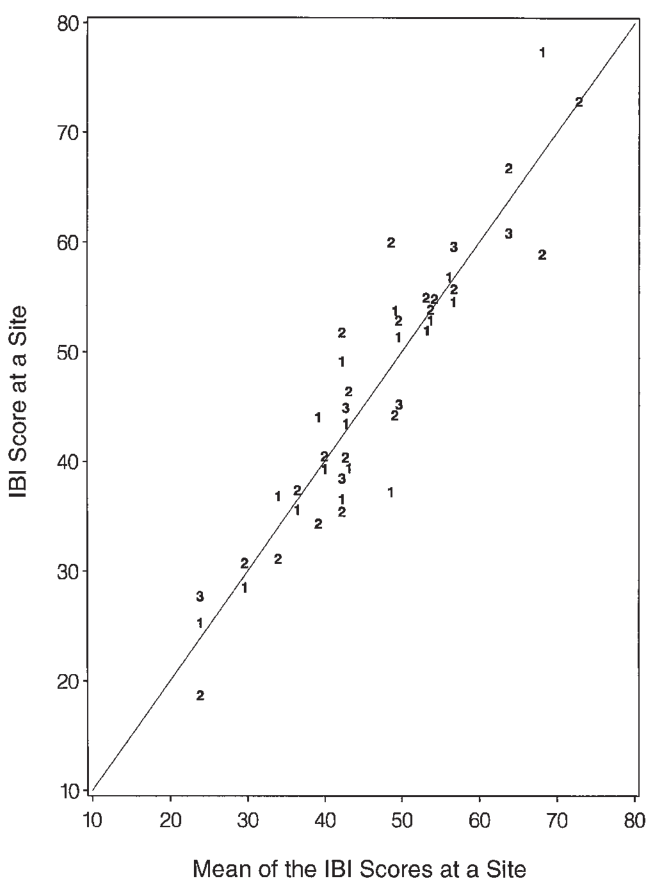


Fig. 5. IBI standard deviation as a function of mean IBI scores at sites sampled multiple times (8 = 1982 sites, 9 = 1992–1995 sites). Lowest mean IBI scores (<40) did not have high standard deviations.

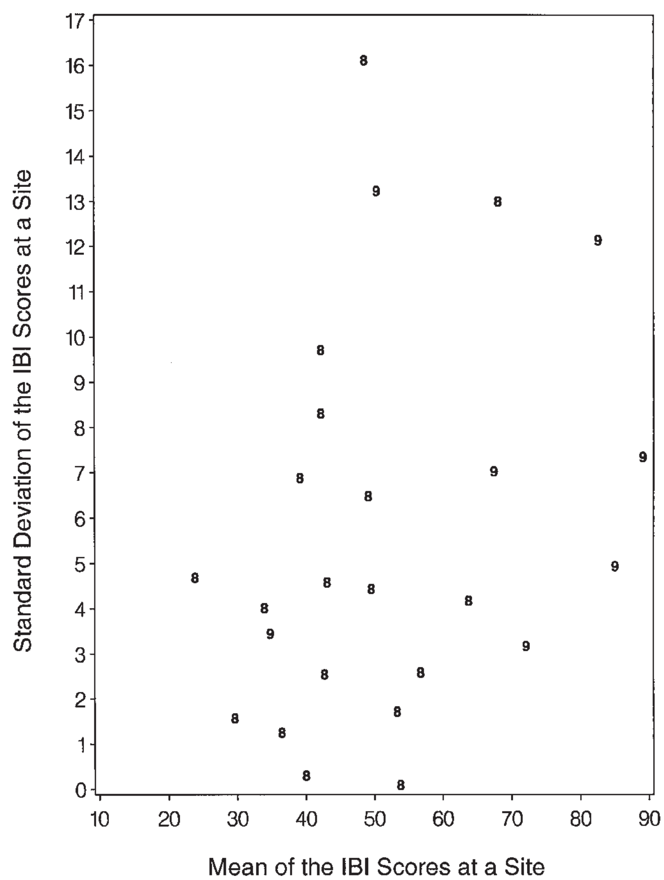
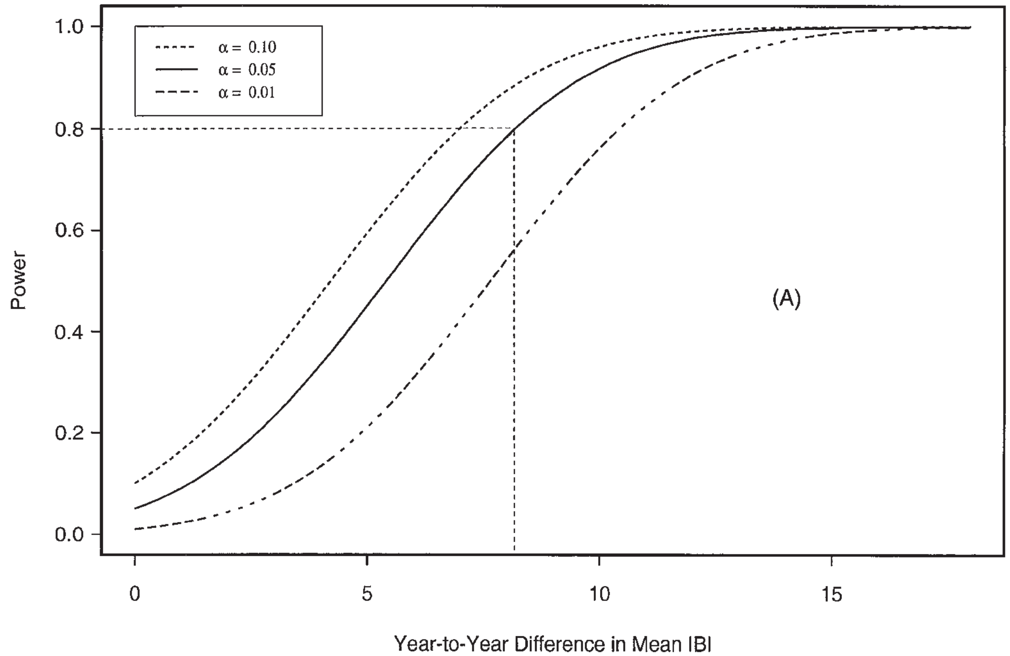
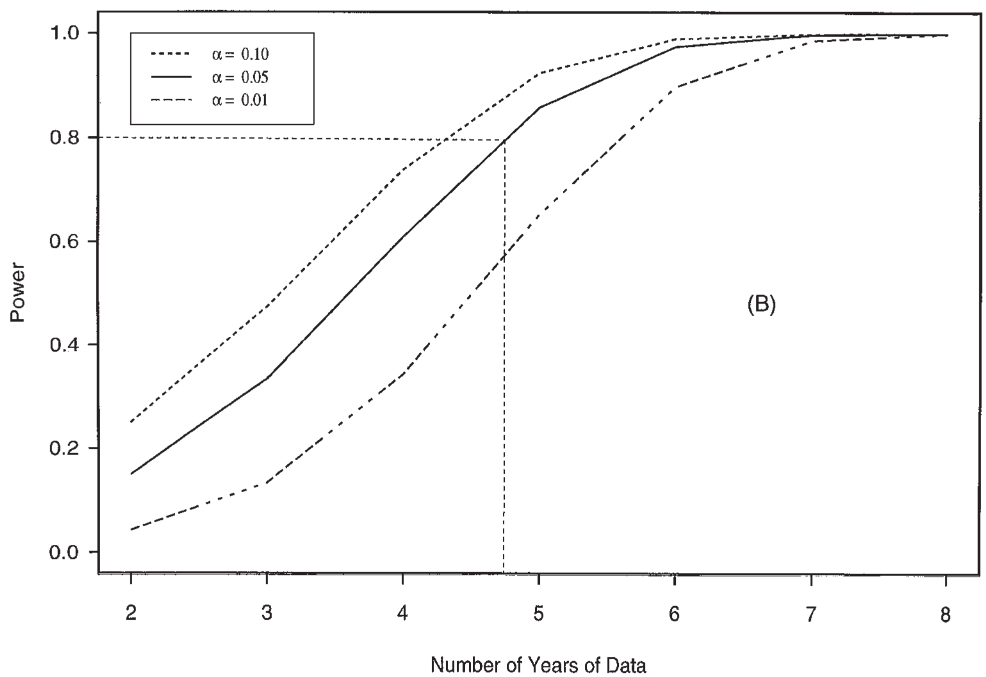


Fig. 6. Power curves for a 13-metric IBI with variances as given in Table 6 and a sample size of 18 streams. As indicated by the straight broken lines intersecting on the solid curve, for power = 0.8 and α = 0.05, this IBI detects (A) a 7.7-unit (8%) year-to-year difference and (B) a 2-unit (2%) per year trend in mean IBI in 4–5 years. A higher power or lower α would require a greater IBI difference for detecting a year-to-year difference (Fig. 6A) or a greater number of years to detect a 2-unit per year trend (Fig. 6B).





Discussion

General IBI approach

Cairns et al. (1993) summarized several desirable indicator characteristics. An appropriate indicator is based on our ecological knowledge and conceptual models of ecosystems; it therefore incorporates elements of structure, composition, and function. It is also useful in waters other than those in which it was developed and is diagnostic, heuristic, or both. Finally, a good indicator has sufficiently small sampling and annual variability to be responsive to marked differences or changes in habitat quality and disturbance levels.

Our index of fish assemblage integrity has all the above characteristics. It was based on our ecological knowledge of the streams and their fishes, and the metrics included aspects of structure, composition, and function. It worked well on a different data set than that from which it was developed, and the individual metrics offered diagnostic and heuristic insights into the major stressors of concern and the probable consequences of continued landscape disturbances. The IBI's sampling and annual variabilities were low enough to allow detection of fairly subtle differences in physical and chemical habitat.

We successfully applied the general approach of Karr et al. (1986) for developing an IBI for streams of an Oregon agricultural region, despite its having no native fish species in common with streams from which they originally developed the IBI. Although we used only five of Karr's original 12 metrics, all but two others (hiders, filter feeders) were derived from the IBI literature. Six original metrics were replaced with more general metrics that can be used in waters where specific species or families are absent or rare. Such changes make IBIs more generally applicable in North America and the world (Hughes and Oberdorff 1998). Several original metrics were initially rejected as unsuitable from knowledge of the local fish fauna; all others were quantitatively evaluated. Candidate metrics that were tested and found highly redundant, unresponsive to physical and chemical disturbance gradients, or highly variable were not included in our final IBI.

Metric and IBI performance

Individual metric evaluations, if performed, are rarely reported. Although there are logical reasons to include each metric, those that are highly variable or unresponsive to physical and chemical habitat quality, like total number of individuals in this case, are best culled from the final index. Another option is to modify the sampling to reduce a source of error. For example, we found that percent filter feeders (lamprey) is particularly vulnerable to sampling error (Table 6), which can be reduced by carefully electrofishing their habitats. Despite selecting more general metrics for use in streams of the Willamette Valley ecoregion and elsewhere, we expect that some of our metrics may be problematic in other regions. We agree with J. R. Karr (University of Washington, Seattle, Wash., personal communication) that IBIs should be based on a set of nationally applicable metrics to the maximum degree possible. Scoring each metric from 0 to 10 and weighting metrics so that the IBI score ranges from 0 to 100 regardless of the number of metrics facilitates comparisons among different IBI scores across the continent or the globe. On the other hand, developing markedly different metrics and IBIs for each basin or ecoregion, instead of modifying scoring criteria or acceptable IBI scores, will likely complicate development and implementation of the IBI by management agencies.

The IBI was strongly correlated with the first principal component of a PCA of IBI metric scores. This suggests that a set of carefully selected metrics can be used in either an IBI or multivariate analysis approach to assess biological integrity, as long as those metrics are not redundant. Both approaches also require evaluation against independent measures of habitat quality or disturbance because IBI and PCA differences among sites may result from purely natural differences in fish species and their relative abundances. Also, both approaches require ecological knowledge; species relative abundances alone are insufficient. The question then is which approach most clearly and easily communicates relative biological integrity to a wide audience.

Our IBI effectively discriminated along disturbance gradients in an independent test using data of Friesen and Ward (1996). The IBI was also significantly correlated with a separate, integrated measure of chemical and physical habitat quality, indicating its sensitivity to a wide range of potential stressors. The relationship of the IBI to the minimum DO concentrations of sites sampled by Friesen and Ward (1996) is also encouraging. It indicates that our IBI was a useful indicator of biological integrity at sites other than those from which it was refined and was associated with DO, a common limiting variable that we did not measure when initially evaluating metric and index responsiveness.

In the development and evaluation of our IBI, we used quantitative biological, physical, and chemical data collected multiple times at multiple sites across an ecoregion. The need for such data may be obvious to many ecologists, but data sets similar to these are too infrequently collected by management agencies. Without multiple visits to a subset of sites, both among years and within the same season, we cannot evaluate critical components of variance and power. We need quantitative physical and chemical habitat data to estimate habitat quality in a meaningful manner and with sufficient precision (Ralph et al. 1994). Finally, we require quantitative, rigorously collected biological data if we intend to precisely assess the biological integrity of stream ecosystems, or to develop useful models of stream performance under varying levels of anthropogenic disturbance. For small Oregon streams, a sampling distance of 40–50 times the mean wetted width is needed to collect 90% of the species obtained by sampling a reach twice that length (L. Reynolds, Oregon State University, Corvallis, Oreg., unpublished data).

IBI statistical power

The variance among stream sites sampled accounted for the preponderance of variance in comparison with temporal and measurement variation (Table 6). This indicates that our IBI discriminated sites of varying biological integrity from the noise of measurement error and short-term temporal variation. Based on the relative sizes of the observed variance components, our IBI can detect an 8-unit (8%) year-to-year difference in mean IBI (for $\beta = 0.2$, $\alpha = 0.05$, $n = 18$) and a 2-unit (2%) per year trend in 5 years (Fig. 6). At the same α and β, Fore et al. (1994) reported a capacity to detect an 8.5-unit difference in mean scores, which is an 18% difference for an IBI with a range of 48 points. Such sensitivity differences between our IBI and that of Fore et al. (1994) may result partly from the larger and more variable database from which theirs was calculated. Our ability to detect such relatively small differences in mean scores reduces the probability of making type 2 errors with the IBI, i.e., failing to detect true differences.

Relationship between IBI variance and IBI scores

Previous researchers reported that IBI variance was greatest at low IBI scores (Karr et al. 1987; Fore et al. 1994; Yoder and Rankin 1995). Others have shown a positive correlation between IBI and its standard deviation, comparable change in IBI score at reference and disturbed sites, or no relationship between IBI score and IBI range or standard error. In our data, standard deviation was greatest at an intermediate IBI score and showed a weak increase with IBI score (Fig. 5). What might account for some of these differences? Karr et al. (1987) compared only two streams and based their covariance conclusion on sites within an individual stream; but they also reported that the stream with the higher quality habitat and IBI scores also had the greater standard deviation. Fore et al. (1994, fig. 6) demonstrated that the preponderance of the variability at one site with low IBI scores was associated with October sampling whereas the comparable summer samples were less variable than those associated with high IBI scores. Yoder and Rankin (1995) used coefficient of variation to portray variability; this statistic results from dividing the standard deviation by the mean. Although it clarifies relative standard deviation, dividing the same standard deviation by a low mean (e.g., IBI of 12) produces a fivefold higher coefficient of variation than dividing it by a high mean (e.g., IBI of 60). Therefore the apparent relationships between variance and low IBI scores previously reported in part result from a misleading statistic, sampling too few sites, or sampling when conditions are more variable. Wherever possible, these sources of variation should be eliminated so that they do not confound IBI variance linked with low or high IBI scores because the degree that a site varies is itself an important aspect of assessing its integrity.

Summary

In this paper, we demonstrated the use of an alternative to regional reference sites for estimating reference conditions and we modified Karr's IBI for use in Willamette Valley streams. Candidate metrics and the IBI were evaluated for responsiveness to varying degrees of chemical and physical disturbance; their variances were examined because studies were designated to estimate major variance components. We produced an IBI responsive to general types of disturbance and capable of detecting an 8% (8-unit) change in mean IBI scores in 1 year and a 2% (2-unit) per year trend in 5 years. This approach for developing and evaluating quantitative indicators to assess the biological integrity of aquatic ecosystems offers a model for indicator development in general.

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